How does the first year tilling a long-term no-tillage field impact soluble nutrient losses in runoff?[☆]

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Abstract

Conservation tillage practices are commonly used to reduce erosion; however, in fields that have been in no-tillage (NT) for long periods, compaction from traffic can restrict infiltration. Rotational tillage (RT) is a common practice that producers use in the central corn-belt of the United States, and could potentially reduce soluble nutrient loads to surface waters. The objectives of this study were to determine the first year impacts of converting from long-term NT to (RT) on N and P losses through runoff. Plots (2 m × 1 m) were constructed in two fields that had been in NT corn–soybean rotation for the previous 15 years. One field remained in NT management, while RT was initiated prior to planting corn in the other field using a soil finisher. Variable-intensity rainfall simulations occurred before and after fertilization with urea (224 kg N ha^{-1}) and triple superphosphate (112 kg P ha^{-1}). Rainfall was simulated at (1) 50 mm h^{-1} for 50 min; (2) 75 mm h^{-1} for 15 min; (3) 25 mm h^{-1} for 15 min; (4) 100 mm h^{-1} for 15 min. Runoff volumes and nutrient (NH$_4$-N, NO$_3$-N and dissolved P [DP]) concentrations were greater from the NT field than the RT field before and after fertilization.

Dissolved P concentrations in runoff prior to fertilization were greater during the 50 mm h^{-1} rainfall period (0.09 mg L^{-1}) compared to the other periods (0.03 mg L^{-1}). Nutrient concentrations increased by 10–100-fold when comparing samples taken after fertilization to those taken prior to fertilization. Nutrient loads were greater prior to and after fertilization from the NT treatment. Prior to fertilization, NT resulted in 83 g ha^{-1} greater NH$_4$-N and 32.4 g ha^{-1} greater dissolved P losses than RT treatment. After fertilization, NT was observed to lose 5.3 kg ha^{-1} more NH$_4$-N, 1.3 kg ha^{-1} more NO$_3$-N, and 2.4 kg ha^{-1} more dissolved P than RT. It is typically difficult to manage land to minimize P and N losses simultaneously; however, in the short term, tillage following long-term NT resulted in lowering the risk of transport of soluble N and P to surface water.
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1. Introduction

Agriculture has been defined as a primary contributor to non-point source pollution of U.S. surface waters (USEPA, 1996). Specifically, nitrogen losses from cropland in the Midwestern U.S. have been cited as a major source of the hypoxic zone in the Gulf of Mexico (Goolsby et al., 2001). Eutrophication of drinking water reservoirs, and estuaries, has been blamed on phosphorus losses from agricultural sources (Schindler, 1977).

Tillage practices can affect nutrient losses to the environment. No-tillage (NT) has been shown to result in greater NH$_3$ volatilization and N$_2$O emissions than conventional tillage (Palma et al., 1998; Venterea et al.,

2005). Leaching of NO_3 has been shown to be greater from NT than conventionally tilled (CT) soil (Brye et al., 2001; Stoddard et al., 2005), however, these results are not consistent in all studies (Stoddard et al., 2005; Thoma et al., 2005). Fertilizer N has been shown to be lost at greater rates in runoff from NT than CT plots (Soileau et al., 1994; Torbert et al., 1996; Francia Martinez et al., 2006). Zhao et al. (2001) observed that practices that produced greater levels of incorporation resulted in lower losses of soluble nutrients. However, in some systems, NT can reduce N losses in runoff (McDowell and McGregor, 1984; Yoder et al., 2005). In a review, Power et al. (2001) concluded that results from NT practices for N were inconclusive.

Incorporation of manure with tillage practices has been shown to be an effective strategy to reduce P losses through runoff (Schreiber and Cullum, 1998; Bundy et al., 2001; Zhao et al., 2001; Little et al., 2005). This observation is likely made because systems without tillage can result in P stratification with high concentrations near the surface, and thus increase the potential for P losses during runoff events (Simard et al., 2000; Sharpley, 2003). On the other hand, at the watershed scale, dissolved P (DP) concentrations in streams were greater with decreasing levels of tillage (Moog and Whiting, 2002). No-tillage has also been shown to result in greater DP concentrations and loads than CT at the plot scale (Daverede et al., 2003). Other studies have shown that CT can increase erosion, and thus total P losses (Torbert et al., 1996; Chambers et al., 2000). One review indicated that conservation tillage practices can result in reducing P losses by 6.1 kg ha^{-1} year^{-1}, and was the most cost effective practice to reduce P losses (Sharpley et al., 2001). Buchanan and King (1993) demonstrated that P losses from plant residues can be increased when they are incorporated into soil instead of left on the surface, such as is the case with NT management.

Most published reports compare the effects of long-term NT or CT systems. However, rotational tillage (RT) management, in which fields are tilled, but not every year, appear to be more a more common practice in the northern corn-belt. Hill (2001) reported that the mean time in continuous NT as of 1999 was 2.4 years in Illinois and 2.3 years for Indiana. Little information has been gathered about how RT or changing from long-term NT to RT impacts nutrient transport. The objectives of this study were to determine the short-term impacts during the first year of converting from long-term NT to RT on soluble N and P losses through runoff. The testable hypothesis used to evaluate these objectives was: There will be no significant difference in soluble N or P transport through runoff from long-term plots in NT or plots recently converted to RT from NT.

2. Materials and methods

Two adjacent fields near Waterloo, IN, USA were used to accomplish the objectives of this study. Both were in a corn/soybean rotation, with corn as the crop planted the year this study occurred (2004). Both fields had been managed using NT for 15 years prior to the study in 2004. The predominant soils at the sites were Glynwood silt loam (Fine, illitic, mesic Aquic Hapludalfs) and Blount silt loam (Fine, illitic, mesic, Aeric Epiaqualfs), and located on a glacial till plain. Slopes in these fields ranged from 2 to 10%. Both fields were approximately 6 ha.

One field remained in NT management. In 2004, tillage was started in the other field, and plans were initiated to perform tillage in this field in the spring, prior to planting, only in years where corn will be planted. On 12 April 2004, a Krause TL 6200 Landsman implement was used to till this field to a depth of 15 cm. This implement had disks, chisel plow shanks, and a harrow, and is commonly referred to as a finisher.

Each field contains a small self-contained catchment that is being monitored for the effects of agricultural management practices on surface water quality. Sites for plots were selected within the management area for each field, but outside of the boundary for the self-contained catchment. Furthermore, location of plots was determined based on topography, such that each plot contained a 5% slope. Three plots measuring 2 m long by 1 m wide were constructed in each field. Plots were hydrologically isolated from the surrounding soil by inserting a 15 cm wide metal border 8 cm deep into the soil. On the downslope edge of the plot, a metal plot-end was used to funnel runoff water into bottles during designated collection periods. Plots within a field were located within 15 m of adjacent plots.

Two rainfall simulations were performed on each plot, with the second occurring 1-week after the first simulation. All rainfall simulations for this study occurred between 22 June and 2 July 2004. Prior to the first rainfall simulation, corn plants were removed from the plots by cutting stalks at the soil surface and disposing of the above-ground biomass. Residue coverage was estimated using the transect method. Prior to the first rainfall simulation, residue coverage was 57% for NT plots, and 20% for RT plots. Approximately 22 cm of natural rainfall occurred between tillage and when rainfall simulations occurred.

During the period rainfall simulations were being conducted only 0.15 cm of rainfall occurred, and plots were covered with plastic during these natural rainfall events. No fertilizer was applied prior to the first rainfall simulation, whereas 1 day prior to the second rainfall simulation, urea was surface applied to plots at a rate equal to 224 kg N ha^{-1} and triple super phosphate was surface applied to plots at a rate equivalent to 112 kg P$_2$O$_5$ ha^{-1}. There was not any natural rainfall between fertilizer applications and rainfall simulations.

Rainfall simulations were performed using programmable oscillating nozzle trough simulators. Veejet 80100 nozzles were spaced 1.1 m apart, and 2.5 m above the soil surface. Rainfall intensity was controlled by varying the frequency of oscillations. Simulations occurred for a duration of 95 min. Rainfall intensity varied in the following order: (1) 50 mm h^{-1} for 50 min; (2) 75 mm h^{-1} for 15 min; (3) 25 mm h^{-1} for 15 min; (4) 100 mm h^{-1} for 15 min. These rainfall intensities were chosen to reflect the intensities commonly observed in natural rainfall events in the study region. The duration of each simulated rainfall intensity was chosen to ensure steady state runoff conditions occurred, so as to minimize the effects of antecedent moisture conditions.

Discrete runoff samples were collected every 5 min for 50 min, after which discrete runoff samples were collected at 3 min time increments. A runoff sample was collected in a 1-L bottle to determine the volume of runoff and mass of sediment for each runoff increment. Sediment data are presented in a companion paper (Warnemuende et al., in press). A 20-mL subsample was collected, filtered (0.45 μm) and acidified (pH < 2.0 with concentrated HCl) for determination of NO$_3$-N, NH$_4$-N and DP. A Konelab Aqua20 autoanalyzer (Thermo Electron Corp. Franklin, MA) was used to analyze NO$_3$-N, and NH$_4$-N colorimetrically. Dissolved P was analyzed using inductively coupled argon plasma (ICAP) spectrometry using a Perkin-Elmer Optima 2000 (Shelton, CT).

Soil hydraulic properties at each of the field sites were measured in situ using the instantaneous profile method (Hillel, 1980). The method involves gravimetric soil sample analysis, double-ring infiltrometry, and tensiometric data analysis. A double-ring infiltrometer with two concentric metal rings having diameters of approximately 90 and 50 cm, respectively, were co-located with tensiometers placed at depths of 15, 30 and 60 cm in the soil profile located just outside the inner ring. The rings were completely filled with water the day before measurements began to pre-wet the soil. On the day of measurement, water was carefully ponded in

the rings with the change in water level over time observed. Once the rate of change became constant, the vertical flux of water in the profile was assumed to be at steady state. At this time the hydraulic conductivity in the zone of constant metric potential is said to be numerically equal to the flux density of water and thus a value of saturated conductivity (K_{sat}) was obtained. Tensiometric readings were taken at this time as a check on unit gradient conditions and saturated water content.

Selected soil physical and hydraulic properties were determined in the laboratory in both fields to a depth of at least 60 cm in 5 and 15 cm intervals. Soil cores were extracted from the site using a soil core-sampling tool having a 15 cm long barrel with a 5 cm inside diameter. Each soil core was divided into 7.5 cm long subsamples. One subsample was used to determine soil texture using the hydrometer method (Day, 1965). The remaining subsample was used to determine the soil water characteristics using the procedure given in Ahuja et al. (1985). Bulk density and θ_v at saturation and at 1, 5, 10, 20, 33, 100, 500, 1000, and 1500 kPa were determined for each 15 cm interval in the profile. Saturated hydraulic conductivity in the laboratory was estimated using an empirical function (a modified form of the Kozeny-Carmen equation) describing K_{sat} as a power function of effective porosity (φ_e). The method is based on the experimental studies of Ahuja et al. (1988), in which effective porosity is defined as saturation water content (θ_s) minus the 1/3 bar water content. The equation is written as:

$$K_s = 764.5\varphi_e^{3.29}$$

where K_{sat} is in cm h^{-1}, and φ_e is given in cm^3 of pores per cm^3 of bulk soil. Values used to estimate K_{sat} were obtained from soil core laboratory measurements of the water characteristic curve. Considering the magnitude of errors involved with field-measured K_{sat} due to the presence of macropores and air entrapment, the proposed equation has shown promise (Ahuja and Hebson, 1992).

Statistical analysis was performed using SAS v 8.0 (SAS Institute, Cary, NC). Means were compared using general linear model procedures, and separation of means was performed using Fisher's least significant difference. Concentrations and mass losses of P and NH$_4$-N were log-normally distributed, and were therefore log-transformed prior to statistical analysis (Neter et al., 1996). An a priori level of $P \leq 0.05$ was established to determine significance. Given that the three plots for each treatment were in the same field, the authors recognize the presence of pseudo-replication, and the fact that pseudo-replication disregards the

assumptions of randomness that are inherent in analysis of variance. Regression analyses were performed using linear and logarithmic functions in SigmaPlot v. 6.0 (SPSS Inc., Chicago, IL).

3. Results and discussion

3.1. Runoff water quality before fertilizer application

Runoff volumes and rates were greater from NT than RT plots (Table 1). The results of field and laboratory studies for the two fields indicate only small differences in saturated conductivity values for both the 0–15 and 0–60 cm depths, with K_{sat} values being slightly higher for the NT field. Average profile K_{sat} values measured in situ for the NT and RT fields were 0.60 and 0.28 cm h^{-1}, respectively. These values compare well with those estimated based on laboratory data where K_{sat} values of 0.41 and 0.16 cm h^{-1} were estimated for the NT and RT fields, respectively. Calculated K_{sat} values for the 0–15 cm depth intervals for the NT and RT fields were 0.55 and 0.21 cm h^{-1}, respectively. Although the data show

slightly higher values of K_{sat} for the NT field, standard deviations are high in both fields due to within field spatial variability and thus, the values are not considered significantly different. All K_{sat} values measured or estimated are within the range of values reported in the literature for these soil types (Rawls et al., 1982). As expected, greater rainfall intensities resulted in greater runoff rates (Table 2). For the lowest rainfall rate (25 mm h^{-1}), runoff was approximately 14 mm h^{-1}, while at the highest rainfall rate (100 mm hr^{-1}), the runoff was approximately 68 mm h^{-1}. Further discussion of runoff quantity and rates can be found in Warnemuende et al. (in press).

Ammonia-N concentrations prior to fertilizer application were 94% greater from NT plots than the RT plots, while NO$_3$-N concentrations were 57% greater in runoff water from the NT plots (Table 1). Greater nitrogen concentrations in runoff from the NT plots were most likely a result of interflow and exfiltration within the plot area that would transport N and be observed as N in runoff. The greater infiltration in the RT plots would likely yield greater vertical transport of N through the soil profile. Other potential explanations

Table 1
Runoff, ammonia, nitrate and dissolved P concentrations in runoff from plots in fields with no-till and conventional tillage management

Treatment	Rainfall (mm)	Runoff (mm)	NH$_4$-N (mg L^{-1})	NO$_3$-N (mg L^{-1})	Dissolved P (mg L^{-1})
Before fertilization					
No-tillage	91.7	61.5a	0.206a	0.846a	0.059a
Rotational tillage	91.7	35.0b	0.106a	0.537a	0.009a
After fertilization					
No-till	91.7	59.5a	11.8a	2.84a	5.95a
Rotational tillage	91.7	42.5a	3.9b	1.08b	2.78b

Values are treatment main effects for tillage, and as such are the mean values for all rainfall intensity values within a tillage and fertilization treatment. (Numbers within a column for the same rainfall event followed by different letters are significantly different at $P < 0.05$.)

Table 2
Runoff, ammonia, nitrate, and dissolved P concentrations in runoff as a function of rainfall intensity for both rainfall events in no-tillage and rotational tillage fields

Rainfall intensity (mm h^{-1})	Rainfall (mm)	Runoff (mm)	NH$_4$-N (mg L^{-1})	NO$_3$-N (mg L^{-1})	Dissolved P (mg L^{-1})
Before fertilization					
50	41.6	11.2bc	0.264a	1.69a	0.091a
75	18.8	13.4b	0.154a	0.51ab	0.022b
25	6.3	4.4c	0.103a	0.96ab	0.040b
100	25.0	20.8a	0.120a	0.44b	0.033b
After fertilization					
50	41.6	16.5a	10.5a	3.06a	7.86a
75	18.8	13.6a	7.1ab	1.16b	3.58b
25	6.3	4.8b	6.7ab	1.90ab	4.18b
100	25.0	18.8a	6.3b	0.79b	2.32b

Values presented are treatment main effects for rainfall intensity, and thus the values presented are the mean for both tillage treatments. (Numbers within a column for the same rainfall event followed by different letters are significantly different at $P < 0.05$.)

for greater N concentration in runoff water from NT than RT is the leaching of N from surface residues present in NT or stratification of nutrients from NT.

Losses of NH_4-N prior to fertilization were not significantly impacted by the rainfall intensity (Table 2). Initial concentrations of NH_4-N in runoff water were approximately three-fold greater in the first several samples collected from NT plots compared to RT plots at the 50 mm h^{-1} intensity, which may suggest that the NH_4-N concentrations were impacted by the available NH_4-N in the soil prior to rainfall, which was initially greater in the NT treatment due to stratification.

Concentrations of NO_3-N were greater during the lower rainfall intensities (50 and 25 mm h^{-1}). Initial NO_3-N concentrations were greater for the RT plots than the NT plots (Fig. 1A). As time increased during the 50 mm h^{-1} rainfall intensity, the NO_3-N concentrations decreased in runoff water from the RT plots and increased for the NT plots. Altering the rainfall intensity resulted in changes in the NO_3-N concentrations almost immediately, with greater intensities resulting in lower concentrations (Fig. 1A). With the exception of the first 2–3 samples taken from the NT plots, there was a strong inverse relationship between runoff rate and NO_3-N
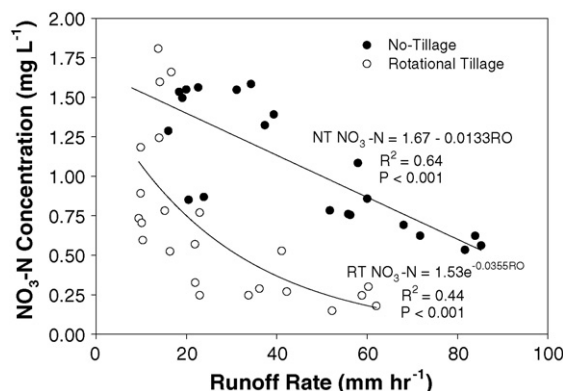
Fig. 2. Nitrate-N concentrations in runoff from rainfall simulations prior to fertilizer application regressed against the flow rate of runoff water throughout the rainfall simulations.

concentration (Fig. 2). A similar relationship was observed for the RT plots; however, the relationship appeared to be described better as a logarithmic decay than as a linear relationship (Fig. 2). This would suggest that when runoff water was being transported off the plots cropped to NT corn, the greater rainfall rates resulted in a dilution of the NO_3-N. However, in RT plots, dilution decreased NO_3-N concentrations during periods of greater runoff intensity, but greater infiltration rates would have also transported more NO_3-N vertically through the profile, and prevented the release of N to surface runoff water from interflow that was exfiltrating the soil.

Dissolved P concentrations were six times greater in runoff from NT plots than the RT plots (Fig. 3A). This was most likely a result of reduced infiltration in the NT plots, and increased concentrations of P at the soil surface. Similar results have been observed when comparing DP among tillage systems (Daverede et al., 2003). No-tillage management can result in the buildup of nutrients in the surface layers, especially when fertilizers are broadcast on the soil surface instead of incorporated into the soil. Furthermore, as runoff water flows across the soil, P can be leached from surface residue and decaying organic matter. Simard et al. (2000) suggested that in pasture systems, P accumulates at the surface, thereby increasing the potential for P losses through overland flow. The stratification of nutrients at the surface would have been reduced as a result of the tillage operation (Hussain et al., 1999; Sharpley, 2003). Organic matter and crop residues would also have been incorporated, reducing the potential for loss of P from these materials into surface runoff.

Dissolved P concentrations in runoff from the first rainfall simulation were significantly greater at the
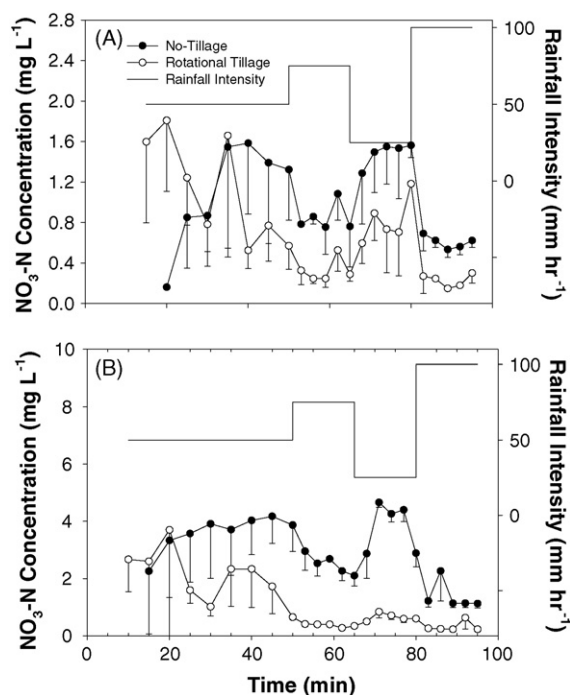

Fig. 1. Mean values of nitrate-N concentrations in runoff water with time and the corresponding rainfall intensities for rainfall simulations before the application of fertilizers (A) and after the application of inorganic N and P fertilizers (B).
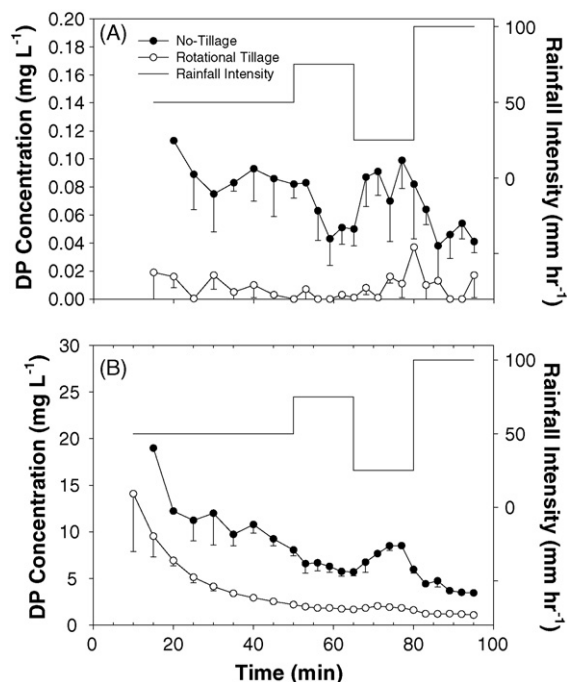
Fig. 3. Mean values of dissolved phosphorus concentrations in runoff water with time and the corresponding rainfall intensities for rainfall simulations before the application of fertilizers (A) and after the application of inorganic N and P fertilizers (B).

50 mm h^{-1} rate than the other three rainfall intensities (Table 2). There was no obvious pattern for DP concentrations from information in Table 2; however, if DP concentrations are regressed against runoff rates, a strong inverse relationship existed for the NT plots, whereas there was no trend apparent from data gathered from the RT plots. As was stated earlier, P concentrations were greater in the NT plots than the RT plots, and with such low concentrations in the RT plots, it was difficult to discern if dilution was a contributing factor to lower P concentrations. While there may have been slight differences in antecedent moisture content prior to rainfall simulations, the protocol used for this rainfall simulation was designed to produce sufficient rainfall to produce steady-state runoff from plots. Thus, runoff results from plots, particularly during steady-state runoff conditions occur are comparable between treatments.

Nutrient loads during the first rainfall event were greater from NT plots than RT plots (Table 3). Loads were more than 150% greater for both NH$_4$-N and NO$_3$-N and more than 10-fold greater for DP from the NT plots. Greater loads would be expected from these plots, due to greater concentrations of the nutrients and the greater runoff volumes.

Table 3
Mass losses of ammonia-N, nitrate-N and dissolved phosphorus in runoff before and after fertilization with inorganic nitrogen and phosphorus fertilizers

Treatment	NH$_4$-N (g ha^{-1})	NO$_3$-N (g ha^{-1})	Dissolved P (g ha^{-1})
Before Fertilization			
No-tillage	120a	514a	35.7a
Rotational tillage	37b	195a	3.3b
After fertilization			
No-tillage	7080a	1750a	3580a
Rotational tillage	1720b	465b	1200b

Numbers within a column for the same rainfall event followed by different letters are significantly different at $P < 0.05$.

3.2. Runoff water quality after fertilization

Runoff volumes and rates were as much as 20% greater from the rainfall simulations after fertilization compared to the volumes and rates observed prior to fertilization (Tables 1 and 2). In both simulations, the NT plots produced more runoff than the RT plots.

Ammonia-N concentrations following fertilization were 10–100-fold greater than the concentrations observed prior to fertilization (Table 1), with NH$_4$-N concentrations from NT management three fold greater than RT. As was previously observed, NH$_4$-N concentrations were greater at the onset of runoff, and were not greatly impacted by rainfall intensities (Table 2). Similar observations have been made following the application of poultry litter (Pierson et al., 2001).

Nitrate-N concentrations were roughly 2–3 times greater in the rainfall simulation following fertilization compared to concentrations prior to fertilization (Table 1). This would suggest rapid nitrification of NH$_3$ released from urea, as fertilizer was added to soil 1 day prior to the second rainfall simulation. As with the previous rainfall simulation, NO$_3$-N concentrations were greater from NT plots than RT plots (Table 1). Runoff concentrations of NO$_3$-N were greater during period of lower rainfall intensities (50 and 25 mm h^{-1}; Table 2). Analysis of discrete samples indicates that NO$_3$-N concentrations decreased after the first few samples from the RT plots, whereas NO$_3$-N concentrations were increased during the same period from the NT plots (Fig. 1B). As suggested previously, the lower concentrations were during periods of greater runoff rates, likely a factor of dilution. As with the previous rainfall simulation, runoff rates were correlated to NO$_3$-N

concentrations, with an inverse linear relationship observed for the NT treatment, and a logarithmic decay relationship observed for the RT. For the RT treatments, the regression coefficient was much weaker ($R^2 = 0.44$ for first rainfall simulation and 0.14 for second rainfall simulation), suggesting that the mechanism of infiltration moving NO_3-N and water vertically through the profile and reducing NO_3-N loading through exfiltration was not as strong. This relationship was also weakened by relatively low NO_3-N concentrations during the 25 mm h^{-1} intensity from the RT plots (between 0.59 and 0.84 mg L^{-1}) compared to the relatively high concentrations that occurred for similar runoff rates at the initiation of runoff (between 1.6 and 2.7 mg L^{-1}).

Mean DP concentrations in runoff for the rainfall simulation occurring after fertilization are presented in Fig. 3B. Dissolved P concentrations in runoff were more than 100-fold greater following fertilization than prior to fertilization (Table 1). Runoff from NT resulted in DP concentrations more than 100% greater than runoff from RT plots (Table 1). As runoff rates increased, DP concentration decreased for both NT and RT plots, however, this was slightly skewed due to the greatest P concentrations occurring during the initial flush of runoff (Fig. 4). As an example, the mean runoff concentration for the first four samples from the RT plots was 8.9 mg L^{-1} whereas the mean concentration for the remainder of the samples was 2.0 mg L^{-1} (Fig. 3B). This is consistent with other studies that reported that practices that increase infiltration can also decrease P transport in runoff (Simard et al., 2000).

As with nutrient loads prior to fertilization, nutrient loads in the rainfall simulation that followed fertilization were greater for the NT treatment than

RT plots (Table 3). Ammonium-N loads were 300% greater in the NT plots, while NO_3-N loads were 275% greater from NT than RT plots. Dissolved P loads were 200% greater for the NT treatment. Dissolved P loads were 100-fold greater after fertilization compared to the rainfall simulation that occurred before fertilization. When comparing the N loads before and after fertilization, NH_4-N loads were more than 10-fold greater and NO_3-N loads were more than 130% greater following fertilization. It should be noted that the loads observed following fertilization represent a worst-case scenario, as the simulated rainfall occurred 1 day after fertilization, which results in the greatest losses of nutrients from both inorganic and organic fertilizers.

4. Conclusions

Ammonium-N and NO_3-N mass loads were more than two-fold greater from NT than RT prior to fertilization and more than three-fold greater from NT following fertilization. Dissolve P loads were more than 10-fold greater and three-fold greater for NT than RT before and after fertilization, respectively. Load data were a result of greater runoff volumes during the first rainfall simulation and greater concentrations in the second rainfall simulation, which occurred 24 h after fertilization. From these data, at least for the short term during the first year, conversion from NT to RT may result in reduced soluble nutrient losses to surface waters.
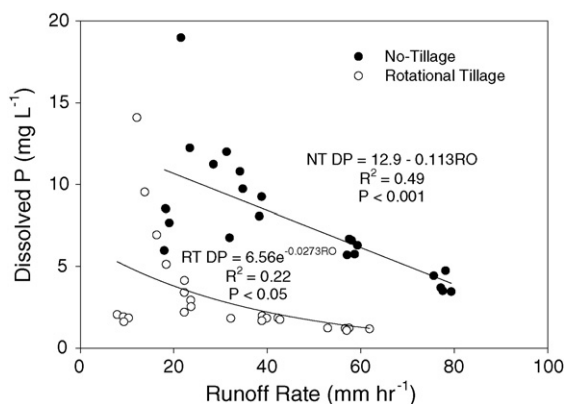
Fig. 4. Dissolved phosphorus concentrations in runoff from rainfall simulations after inorganic fertilizer applications regressed against the flow rate of runoff water throughout the rainfall simulations.

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